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A literature-based multi-criteria evaluation of the EU ETS

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ABSTRACT

This article reviews the existing literature on the European Union Emission Trading Scheme (EU ETS), focusing on empirical ex-post research since the end of the first period (2007). The literature is presented through a multi-criteria evaluation. Concerning environmental effectiveness, despite over-allocation during the first period, abatement is estimated between −2.5% and −5%. Trade-driven carbon leakage was not observed, even if long-term economic models predict divergent leakage estimates for certain at-risk sectors. The abatement target was likely to be below an economically efficient level, but was reached in a fairly cost-effective way, even if free allocation gave rise to several distortional effects. Equity concerns were manifold and constitute a major drawback to the policy. Finally, institutional feasibility can be considered positive in that the EU ETS passed the European legislative process, unlike the previously proposed EU-wide carbon tax.

Contents

Abbreviations: BAU, business as usual; CDM, Clean Development Mechanism; CGE, Computable General Equilibrium; EBIT, earnings before interest and taxes; EBITDA, earnings before interest, taxes, depreciation and amortisation; EU ETS, European Union Emission Trading Scheme; IPCC, Intergovernmental Panel on Climate Change; JI, Joint Implementation; MAC, marginal abatement cost; MRV, monitoring, reporting and verification; NACE, Nomenclature statistique des Activités économiques dans la Communauté européenne (acronyme used in English); NAP, National Allocation Plan; NGO, Non-Governmental Organisation; SCC, social cost of carbon; UNFCCC, United Nations Framework Convention on Climate Change

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1. Introduction

The European Union Emission Trading Scheme (EU ETS), launched in 2005 can be considered as the cornerstone of the EU climate policy, attracting the attention of many researchers and policy makers, in Europe as well as in other regions of the world. The EU ETS is by far the largest environmental market in the world, covering 27 Member States and over 11,500 plants from the electricity, combustion (>20 MW capacity), coke, iron, steel, cement, lime, glass, ceramics, brick, tile, refinery, paper and pulp industries. The ETS sectors' CO_2 emissions account for approximately 40% of total greenhouse gas emissions in the European Union.

There is an urgent need for a structured review of the existing literature because the EU ETS is highly debated and opposing opinions continue to fuel controversies among scholars and policy makers as well as NGOs. The aim of this review is to delineate the strong and weak points of the EU ETS based on an overview of the academic literature since 2008, when data and empirical results for the first period became available. The methodology of our multi-criteria approach is explained in Section 2. Sections 3, 4, 5 and 6 asses the environmental effectiveness, economic efficiency, distributional effects and institutional feasibility of the EU ETS. Section 7 concludes. Since most polemics require a good understanding of the legal design of the ETS, the following paragraphs describe the main features of the ETS from its start until 2020.

1.1. 2005–2007: 1st period of the EU ETS

In 2000, the European Commission launched its Green Paper on emissions trading [1]; in 2003 the European Parliament and the Council passed the ETS-Directive [2] and on 1 January 2005, the system came into force.

Every Member State had its own National Allocation Plan (NAP) that determined the number of grandfathered (free) allowances allocated to each plant according to national criteria. Despite the Commission's corrections to 15 NAPs that lowered the total cap by 290 million tonnes per year (-4.6%) [5], allocations exceeded verified emissions every year, leading to a price crash in April 2006 (Fig. 1).

As production units received their allowances in the beginning of the year and had to introduce them before the end of April the following year, borrowing was possible with a maximum of one year of allowances. Banking between the different years of the period was allowed, but not between the first period and the second.

No more than 5% of the allocations per country were allowed to be auctioned [2], but only Hungary, Ireland and Lithuania

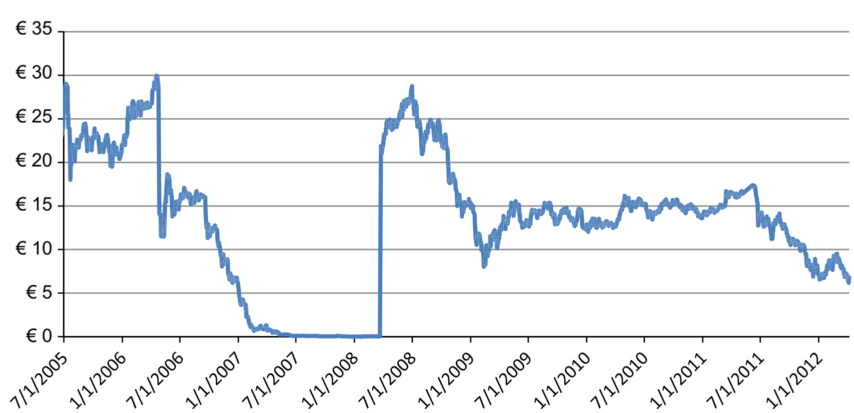

Fig. 1. Evolution of the carbon price (EEX).

actually held auctions. Denmark had planned to auction 5% of its allowances, but decided to sell permits through the brokered market when prices fell to €0.90–2.20 in February 2007 [3]. Only 0.2% of total European allowances were sold [4].

1.2. 2008–2012: 2nd period

The second phase of the ETS corresponds to the period of the Kyoto protocol. Initial allocation through NAPs of Member States yielded a total cap of 2325 million allowances, largely surpassing the cap of the first phase. After revision by the Commission, NAP allowances were lowered by 245 million tonnes of emissions per year (−10.4%) [5]. The caps of only four countries (Denmark, France, Slovenia and the UK) were not revised [4]. The Commission's revision led to a decreasing cap with an overall target of 6.5% decrease in emissions compared to the 2005 level. Fig. 2 shows the decline in allowances between the first period and the second.

In this second period, Member States are allowed to auction up to 10% of their allowances, but the NAPs foresee auctioning only 3.1% of total allowances [4]. This mainly concerns Germany and the UK [4], but 9 other countries have planned smaller auctions. In order to temper price volatility at the end of the period, banking between the 2nd and 3rd phase will be allowed.

The Kyoto Protocol created project-based offsetting markets: Clean Development Mechanisms (CDM) in developing countries and Joint Implementation (JI) in economies in transition. In the 2nd period, these 'Kyoto credits' are accepted for compliance in the European market. For each production unit (or sector), the NAPs determine the maximum share of Kyoto credits for compliance. The European Commission lowered the maximum use of Kyoto credits from 374 million per year, as proposed in the initial NAPs, to 274 million per year [4].

Peaking at €27 in July 2008, the price for ETS allowances dipped below €10 at the beginning of February 2009 due to the economic downturn. Between April 2010 and April 2011 the price was relatively stable, around €15. The European sovereign debt crisis, pushed the carbon price further down to 7€ in April 2012 (Fig. 1).

Since the beginning of 2012, all airlines landing in European airports are integrated in the ETS, which will add an additional sector to the existing ones concerned by the ETS. The allowances for the aviation sector are 97.5% of their 2004–2006 emissions. These allowances will be grandfathered at 85% [6].

1.3. 2013–2020: 3rd period

During the 3rd phase of the ETS, aluminium, mineral insulation wool, several chemicals and carbon capture & storage will be integrated into the system. Allowances will no longer be determined by NAPs, but based on product benchmarks. The benchmark is the emissions per ton by the 10% most carbon efficient companies in the EU. Overall allowances (except the part allowed for the aviation sector, which will remain stable at 95% of 2004–2006 emissions) will decrease by 1.74% per year starting in 2010, resulting in a 21% reduction by 2020 compared to 2005 emissions. The EU commission targets 50% of allowances to be auctioned in 2013, with a gradual increase of auctioning, reaching 100% in 2027. Free allowances will be allocated as follows:

- Electricity producers will no longer receive free allowances, except for highly efficient cogeneration and district heating.
- Sectors considered to be exposed to carbon leakage (increase in emissions abroad because of increased net importation of carbon-intensive goods) will receive 100% of the benchmarked allocation for free. The other sectors will receive 80% of the benchmarked allocation for free in 2013 with a gradual decrease of free allocation to 30% in 2020.
- The aviation sector will continue to receive free allocation at 85% [7].

2. Methodology

The multi-criteria framework in this study is founded on the premise that various criteria are needed to evaluate a policy. Evaluation is by nature normative and thus some specific criteria must be utilized as a basis for these normative judgements [8]. A set of value criteria – a normative framework – is needed to make a global assessment of a policy. A panoply of evaluation criteria exists, and different authors all use different criteria to assess environmental policies [8–17]. Few guidelines are available to choose the most suitable criteria for environmental policy [12,14]. The 4th assessment report of the IPCC [14, p. 751] distinguishes four principal criteria for evaluating environmental policy instruments:

- Environmental effectiveness—the extent to which a policy meets its intended environmental objective or realizes positive environmental outcomes.
- Cost-effectiveness—the extent to which the policy can achieve its objectives at a minimum cost to society.
- Distributional considerations—the incidence or distributional consequences of a policy, which includes dimensions such as fairness and equity, although there are others.
- Institutional feasibility—the extent to which a policy instrument is likely to be viewed as legitimate, gain acceptance, adopted and implemented.

This study follows these criteria because they best fit the practical aims of EU ETS and are recurrent in other environmental

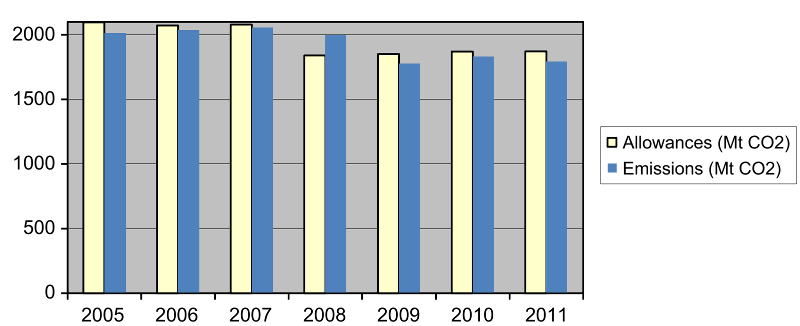

Fig. 2. Allowances and verified emissions for 2005–2011, ETS sectors, EU-25. (CITL on 03/04/2012 available from http://www.dataservice.eea.europa.eu/PivotApp/pivot.aspx?pivotid=473 and www.carbonmarketdata.com for year 2011). N.B.: Romania and Bulgaria, who joined the EU ETS in 2007, are not included in the figure in order to keep data comparable. Note however that when adding Romania and Bulgaria, the total allowance surplus for the second period quadruples from 33 Mt CO_2 for EU 25–134 Mt CO_2 for EU 27.

policy evaluation studies [8–12,16,17], even if these studies include additional criteria. According to Stavins [[17], p. 303], the first three criteria are particularly important for the assessment of a domestic climate change policy. The criteria mentioned above are also justified by hypotheses put forth in the public arena for the EU ETS. The European Commission [1] launched the EU ETS, arguing that it was environmentally efficient and cost-effective. Distributional effects of the EU ETS – be it between individual companies, countries, consumers and producers, the public and private sectors, low and high incomes – are highly controversial. And since the attempt to instate an EU-wide carbon tax during the beginning of the nineties resulted in failure, institutional feasibility was the reason why the EU ETS became a reality rather than its closest policy alternative.

Each criterion is evaluated with respect to various aspects or sub-criteria that were selected according to highly debated questions in the EU ETS literature and recurrent sub-criteria in the previously mentioned multi-criteria evaluation studies.

Concerning cost-effectiveness, the IPCC [[14], p. 751] acknowledges that general economic efficiency encompasses more than cost-effectiveness. Cost-effectiveness is distinct from general economic efficiency. Whereas cost-effectiveness takes an environmental goal as given, efficiency involves the process of selecting a specific goal according to economic criteria. As there is a huge branch of literature evaluating the economic efficiency of climate policy and as the goals of the EU ETS have been criticized for not being ambitious enough, general economic efficiency is evaluated too.

Academic research on the EU ETS has developed rapidly over the last years, since it is by far the most important experiment in pollution trading schemes in the world. The EU ETS is a complex policy covering 27 countries involved in international negotiations and thousands of companies acting on international markets, so papers based on different philosophies, paradigms and axiologies obtain different conclusions [18]. That is why an evaluation based on a literature review is of primary importance. Such a literature review will be helpful as a first step in a multi-criteria decision making process, where decision makers choose a policy or energy system among a set of alternatives by quantifying preferences over a number of (conflicting) objectives. For overviews on multi-criteria decision making for energy systems, see [19–21].

This article is based on 115 academic studies on the EU ETS, which makes it possible to discern questions that raise a consensus from questions that raise controversy. This enhances the external validity of the approach [18]. The focus is on recent research, published since the end of 2007, when empirical work based on EU ETS data came out. In order to go through the literature systematically, the databases Science Direct, Econlit, BusinessElite, Springer and Wiley were searched for the keywords "EU ETS" or "Europe* emission trading" in titles or abstracts between 2008 and 2010. All studies related to one of the 4 evaluation criteria were included. Based on the references of these studies, the most relevant studies published before 2008 were also included.

3. Environmental effectiveness

3.1. Abatement

As the EU ETS was set up to mitigate emissions from the industries concerned and to reach the European Union Kyoto target, environmental effectiveness is determined in the first place by the level of abatement.

Abatement is defined as the difference between verified emissions and BAU (business-as-usual) emissions, which are difficult to estimate. Moreover, abatement is difficult to estimate for the first phase of the EU ETS because there are no aggregated data for emissions of ETS plants before 2005.

Ellerman and Buchner [22] estimate abatement for 2005 and 2006 based on the following counterfactual:

$$BAU_t = emissions\,from\,NAPs_{2004} \times \frac{GDP_t}{GDP_{2004}} \times carbon\ intensity\ \ trend_{2000-2004}^{t-2004} \quad (1)$$

Historic emissions are derived from data found in NAPs of Member States, but these data have two problems: potential bias and imperfect comparability. Potential bias arises from the fact that data collection was largely a voluntary submission by the companies involved and it was conducted under severe time constraints, limiting possibilities for the authorities to check the data. Imperfect comparability refers to the fact that the estimates of different NAPs were calculated using different methodologies. Compared to their counterfactual, Ellerman and Buchner [22] estimate the EU23 emission abatement to be -3.1%, *i.e.* -2% for EU15 and -7.7% for Eastern Europe. After a consistency check based on output and GDP growth combined with trends in carbon intensity, they estimate abatement between 50 Mt (-2.4%) and 100 Mt (-4.7%) in 2005 as well as in 2006.

Ellerman, Convery and de Perthuis [23] used a slightly different methodology, based on the following counterfactual:

$$BAU_t = UNFCCCemissions_{2004} \times \frac{verified\ ETS\ emissions_{2005-2006}}{UNFCCC\ emissions_{2005-2006}} \times \frac{GDP_t}{GDP_{2004}} \times Carbon\ Intensity\ Trend_{2000a2004}^{t-2004} \quad (2)$$

UNFCCC data (United Nations Framework Convention on Climate Change) are not affected by the two problems discussed above but it is not possible to isolate the installations included in the ETS. So a conversion factor derived from the period with overlapping data was used to estimate historic ETS emissions. This study estimates a comparable cumulative abatement to be 210 million tonnes (-3.5%) over the entire first period. Note that the carbon intensity trend was -0.98%/year for the 2000–2004 period and -3.24% for the 1995–2000 period. Extrapolating the trend for a longer period over the last 10 years would have thus led to a lower abatement estimate. This is also true for the estimation by Ellerman and Buchner [22] and Ellerman and Feilhauer [24] (Table 1).

Anderson and Di Maria [25] constructed a BAU counterfactual established using Eurostat aggregate emissions per sub-sector based on NACE classification codes that approximately match the ETS sector:

$$BAU_t = \frac{verified\ \ emissions_t}{eurostat\ \ emissions_t}(\beta_1 emission_{t-1} + \beta_2 manufacturing_t + \beta_3 energy_t + \beta_4 hdd_t + \beta_5 cdd_t + \beta_6 rain_t + \beta_7 elec_t) \quad (3)$$

where $emission_{t-1}$ is the log lagged eurostat emissions, *manufacturing* is the log output of manufacturing sector, *energy* is the log output of energy sector, *hd* is the log annual heating degree days, *cdd* is the log annual cooling degree days, *rain* is the log annual rainfall (measuring hydropower potential), *elec* is the log electricity prices (affecting electricity demand).

For countries where verified emissions are below the BAU counterfactual, the authors conclude a total abatement of 247 Mt of CO_2 during the first phase. However, several countries have emissions that are higher than the BAU counterfactual, which indicates 73 Mt emission inflation during the first phase. Companies may inflate emissions, speculating that they might obtain more grandfathered emissions in the future. The net effect of abatement and inflation is that the verified emissions are 174 Mt below the BAU counterfactual

Table 1
Estimated abatement during the first phase of the EU ETS according to several studies.

Authors	Estimated abatement	Sector & country	Methodology
Ellerman and Buchner [22]	50–100 Mt/year (2.5–5%)	All sectors in EU	BAU counterfactual based on NAPs
Ellerman et al. [23]	70 Mt/year (3.5%)	All sectors in EU	BAU counterfactual based on UNFCCC emissions
Anderson and Di Maria [25]	58 Mt/year (mean) (3%)	All sectors in EU	BAU counterfactual based on Eurostat aggregate emissions
Delarue et al. [26]	59–88 Mt/year	Electricity in EU	Fuel switch model
Ellerman and Feilhauer [24]	29 Mt/year (6%)	All sectors in Germany	BAU counterfactual based on NAP
	25 Mt/year (5%)		BAU counterfactual based on UNFCCC
	4.4 Mt/year	Electricity in Germany	Fuel switch model
McGuiness and Ellerman [27]	13–21 Mt/year	Electricity in UK	Fuel switch model

(−2.8%). The estimated abatement is comparable (slightly lower) than the abatement found by Ellerman et al. [23] and Ellerman and Buchner [22].

Results of different abatement estimates, also including sectorial or national estimates, are summarized in table 1. Most studies estimate emission abatement between −2.5% and −5% for the first phase of the EU ETS. Note that if the Commission had not imposed 4.6% reduction of allocation in initial NAPs, there would have been no abatement.

Several qualitative studies find contrasting abatement ambitions during the first phase of the EU ETS. Based on a survey of over 300 ETS firms in Germany, the UK, Denmark and the Netherlands, Engels [28] reports that about one-third of firms surveyed did not know the abatement costs within their own companies. Engels concludes a level of ignorance among some of the companies, showing that they did not use phase 1 to develop a carbon strategy. On the other hand, the majority of companies placed the responsibility for emissions trading on the top level of the organisational hierarchy in 2005, suggesting that most companies did indeed develop a carbon strategy over time.

Sandoff and Schaad [29] find in a survey of 114 Swedish firms that only 37% of firms had concrete abatement objectives for the reduction of CO_2 emissions, which may indicate low abatement ambition. On the other hand, just like Engels [28], they found that (top) management played a prominent role in allowance management, indicating that the EU ETS has considerable weight and status within many organisations.

The evolution of verified emissions of the ETS sector can be seen in fig. 2, which shows a steep decline in emissions during the second phase. Again, a comparison against a business-as-usual scenario should be made to evaluate abatement, because the economic crisis partly explains the emission decline. There is, however, no peer reviewed study on the phenomenon. But as the cap is 6.5% lower than the 2005 emissions, we can conclude that emissions, that were already reduced during the first phase, continue to decrease during the second phase. The extent to which the speed of abatement is in accordance with future climate damage will be discussed in Section 4.1.

3.2. Over-allocation

Over-allocation has an impact on several evaluation criteria. First of all, over-allocation reduces the abatement level. This is why over-allocation is discussed here as part of the environmental effectiveness. Note however that over-allocation also indicates a problem of transparency, because in order to justify over-allocations, Member States report biased historic emission estimates and over-optimistic business-as-usual projections in National Allocation Plans. Moreover, over-allocation entails distributional effects between companies, sectors and countries (see further). Finally, over-allocation decreases predictability because when there is over-allocation, the cap on emissions, fixed by the regulator, does not lead to the estimated abatement effort it was meant to enforce (see further).

The overall allocations to the ETS installations during the first 3 years exceeded their verified allowances every year (fig. 2). This led to a price crash at the end of April 2006 when the first verified emissions were published. The spot price collapsed from €29.20/t CO_2 on Monday, 24 April, to a closing spot price of €13.35/t CO_2 at the end of that week. Since banking from one period to another was not allowed, the overall market was long in allowances, which led to the further collapse of the price, ending at €0.08/t CO_2 by the end of 2007.

Over-allocation can be defined in various ways. A long position of a country (allocation exceeding verified emissions) can be explained by over-allocation or else, on the contrary, by greater-than-expected abatement due to underestimated abatement opportunities, weather events, shifting gas/coal price ratios… By definition, a market supposes long installations that sell to short installations. The same applies for aggregate data per country or per sector. Thus a long position for a country or sector does not necessarily imply over-allocation.

Anderson and Di Maria [25] define over-allocation as an allocation above BAU emissions. They estimate that 6% of free allowances were above BAU emissions during the first period.

Ellerman and Buchner [22] determine over-allocation based on the ratio between short and long positions of individual companies per country. They consider that when the sum of short positions is less than 40% of the sum of long positions, a country has over-allocated its companies. For the 2005–2006 data, they find that this is the case for 11 countries, who distributed 30% of total EU allocations. Their over-allocated surplus is 6% of total allowances or 125 million European Union Allowances (EUAs) per year.

Clo [30] defines over-allocation as a mitigation effort made by the ETS sectors to attain the Kyoto target, which is less than proportional compared to the non-ETS sectors. The Kyoto emission target sets the sum of abatement for the ETS and non-ETS sectors in 2008–2012 for each country. If marginal abatement costs for both categories were the same, a proportional effort for the ETS sectors compared to the non-trading sectors would constitute a cost-effective solution. The pre-2005 share of ETS emissions is estimated to be between 41% and 43%. The cap for the first period for the EU 23 was 2172 Mt CO_2, almost 44% of the virtual EU-23 Kyoto target. This indicates over-allocation [30].

Concerning the second period, 2008–2012, the cap that was finally approved (1955 Mt or 39.5% of the Kyoto target), does not indicate over-allocation [30] and is in accordance with the idea of larger cost-effective potential, in particular in the electricity sector, compared to non-ETS sectors [[31], p. 9]. The initial proposal in NAPs from Member States, before refusal by the Commission, would have assigned 43.5% of the Kyoto target to the ETS sectors and clearly would have resulted in over-allocation [30].

Schleich et al. [4] also conclude for the 2nd period that over-allocation did not occur. On average, after correcting for

installations that were allowed to be opted out in the first phase, the cap in phase 2 is an estimated 12.8% lower than emissions in 2005 and corresponds to a proportional effort compared to non-ETS sectors. The net short position of the ETS market in 2008 also indicates no over-allocation [32].

The maximum of CDM and JI allowances for the EU is 273 million per year for the 2008–2012 period. This amount was initially 373 million [4]. This ceiling of 273 million Kyoto allowances allowed, the verified emissions inside the EU during the 2nd phase to exceeded the 2005 emissions, which was in contradiction with the principle of the Kyoto Protocol, that the major abatement effort has to be made inside the country. As 214 million Kyoto credits have already been introduced between 2008 and 2010 [own calculations for www.carbonmarketdata.com], and since the economic downturn has decreased emissions to 133 Mt below the allocations for 2008–2011 period [Fig. 2], a considerable amount of allowances will be banked into the 3rd period, further exercising a downward pressure on carbon prices.

There is a wide consensus that over-allocation occurred during the first period, unlike the case with the revised cap for the second period. However, over-allocation does not mean that there was no net abatement. As 378 million EUAs [33] were never used for compliance (the market as a whole was net long during the first period) and as other countries have set ambitious targets, over-allocation did not inhibit net abatement.

3.3. *Predictability of environmental impact*

The environmental predictability of the ETS is high in the sense that the total cap of emissions is set by the regulator. This relative certainty about the environmental outcome is lauded in the controversy over the choice between a cap-and-trade system and a carbon tax. Both policies are based on the same economic incentive. Under a cap-and-trade system, the high predictability of the environmental outcome has an intrinsically linked counterpart: low predictability of the abatement effort (that is determined by the carbon price). Unlike cap and trade, a carbon tax sets the abatement effort (carbon price) and leaves the environmental target uncertain because under a carbon tax total emissions depend on unknown abatement costs and the price elasticity of demand.

The seminal paper of Weitzman [34] argues that a carbon tax (setting the carbon price) is more appropriate under uncertainty than a cap-and-trade policy (setting total emissions) when the marginal climate benefit curve is relatively flat and the marginal abatement cost curve is relatively steep. The idea is that it is better to control the factor with the highest impact.

Several authors advocate automatically adapting carbon taxes [35–37] or instating a cap-and-trade system with a price floor and price ceilings [16,38], in order to get the advantages of each of these two policies.

Abatement effort uncertainty under ETS can take problematic proportions in specific circumstances. For example the recent economic crisis halved carbon prices between 2008 and 2009, with a minimum of €8.02/tonne on 11 February 2009 and a second dip below €7/tonne since March 2012 (Fig. 1). Also, during the first phase, a price crush was caused by biased historic emissions. In effect, certainty of future emissions does not imply certainty of future abatement, because abatement is defined as the difference between actual emissions and business-as-usual emissions that depend on the economic activity and accuracy of past emissions.

Linking ETS with Kyoto-offsetting credits creates a source of uncertainty as to emissions inside the European Union, but if CDM and JI projects entail a fully additional mitigation then global environmental impact is not affected. Additionality of greenhouse gas reductions of CDM and JI projects is controversial, however, but this discussion goes beyond the scope of this article.

3.4. *Environmental side effects: carbon leakage*

Carbon leakage is the increase of carbon emissions outside the EU as a consequence of climate policy inside the EU. Leakage can arise when, in countries with emission limitations, energy-intensive and trade-exposed industries lose competitiveness, thereby increasing emission-intensive production abroad. This trade channel of leakage is hotly debated because it is closely linked with the competitiveness of ETS industries. Competitiveness also affects political acceptability [10], which will be discussed in this study as the 4th and final criterion.

Insofar as carbon costs are passed through in sales prices, this creates a market share loss (higher imports and lower exports). To the extent that carbon costs are not passed through, the resulting lower profit margin may compromise long-term European investments in new producing capacity.

Leakage should not be confused with the shift of relative competitiveness from polluting products in favour of less polluting substitutes, which is part of the desired result of climate policy. Ambitious climate policy therefore implies long-term incentives for low-carbon innovations, providing first-mover advantages and preparing for a new playing field [39].

There is also "fossil fuel channel" leakage or "energy price channel" leakage when ambitious climate policies within Europe affect world prices for oil, gas and coal and lead to higher consumption of fossil fuels in the rest of the world. This leakage channel, was found to be much more important than the trade leakage channel by certain models [40,41], while of minor impact in other models [42].

Next, several authors [43–46] consider technology spill-overs as a form of leakage. Low-carbon technologies that are developed under stringent climate policy will diffuse to other countries, reducing emissions abroad. Although this effect is likely to be significant [44], it is very difficult to quantify because of the high level of uncertainty about technological development. Technology development also has an effect on competitiveness as European low-carbon technologies create a long-term positive competitive advantage for European producers [46]. Technology development induced by the ETS will be discussed in greater detail under the efficiency criterion.

Finally, there is interdependence between the climate policy of the EU and climate policy of other countries [45]. The Kyoto agreement acknowledges that stringent action in industrialised countries is a prerequisite for future abatement policies. This interdependence is generally not included in the definition of carbon leakage but likely to be important. The risk of provoking distrust in the international negotiations is one of the reasons why the EU actually refrains from border tax adjustments [47,48]. The detailed discussion about the political interdependences in the negotiation of the post-Kyoto agreement falls beyond the scope of this study.

3.4.1. *Ex ante analysis*

Different recent *ex ante* analyses are summarized in Table 2.

Hourcade et al. [49] calculated the cost increases from both €10/MW extra electricity cost and from direct carbon emissions auctioned at €20/t CO_2 for 164 sectors in the UK. They find that only 20 sectors of the 159 manufacturing sectors exceed a direct + indirect cost increase of 4% of gross value added (GVA). The share of these 20 sectors in GDP and employment are respectively 1.1% and 0.5%. Even for an economy that is more based on manufacturing (like Germany, for example), the GDP

Table 2
Ex ante estimates of carbon leakage and competitiveness loss.

Authors	Carbon price (€/t CO_2)	Policy	Cost pass-through	Competitiveness loss induced by ETS and carbon leakage
All sectors				
Manders and Veenendaal [52]	€27/t	Full auctioning	Close to 100%	ETS production −1.7% Economy-wide carbon leakage 0.3%
Bernard and Vielle [53]	From €7/t in 2010 to €71/t in 2020	Full auctioning	Close to 100%	Economy-wide carbon leakage 7%
Kuik and Hofkes [42]	€20/t	Full auctioning	Close to 100%	Iron&steel production −2% Mineral production −1.6% Economy-wide carbon leakage 10.8%
Böhringer et al. [40]	Approximately €45/t	Full auctioning	Close to 100%	Energy-intensive production −5.5% Economy-wide carbon leakage 37%
Steel				
Demailly and Quirion [55]	€20/t (mean)	57% free allocation	75% in EU 50% for export Exogenous	EBITDA constant EU production −1% compared to 2005 Imports +2.5% Exports −2%
Hourcade et al. [49]	€30/t	Full auctioning	50% Exogenous	EBIT from 15% to 10% Import ratio from 17% to 18%
Kuik and Hofkes [42]	€20/t	Full auctioning	Close to 100%	EU production −2% Imports +9% Exports −8% Carbon leakage 35%
Smale et al. [56]	€15/t	100% free allocation	65% Endogenous	EBITDA +12% Output −2.1%
FitzGerald et al. [54]	n/a	n/a	US price has significant influence on domestic price in 5 out of 7 countries	Leakage exposure high Low cost pass-through Low scope for energy efficiency High share of energy expenditures
Cement				
Hourcade et al. [49]	€30/t	Full auctioning	50% Exogenous	EBIT negative Import ratio from 8% to 11% (+37%)
Ponssard and Walker [57]	€20/t	Full auctioning	67% Exogenous	Import ratio from 10% to 20% (+100%)
Smale et al. [56]	€15/t	100% free allocation	'High' Endogenous	EBITDA +12% Output −2.1%
FitzGerald et al. [54]	n/a	n/a	US price has insignificant influence on domestic price in all 7 countries	Intermediate to low: High cost pass-through Intermediate scope for energy efficiency High share of energy expenditures
Chemical industry				
Tomás et al. [60]	€15/t	Full auctioning	n/a	Cost +0.6% to +2.26%
FitzGerald et al. [54]	n/a	n/a	US price has significant influence on domestic price in 2 out of 7 countries	High to intermediate: Intermediate cost pass-through High scope for energy efficiency Very high share of energy expenditures
Refinery				
Reinaud [61]	€10/t	Full auctioning	n/a	Costs +15% to +30% of running expenses
Aluminium				
Smale et al. [56]	€15/t	No compensation for electricity price increase	n/a Endogenous	Closure of UK plants
Aviation				
Anger [62]	€20/t	85% free allocation	Close to 100%	No impact on flights No impact on GDP Aviation emissions −3.4% EU Emissions +0.24% because of revenue recycling.

share of carbon-intensive sectors does not exceed 2% [50]. Moreover, non-price aspects restrict leakage concerns for some sectors: local resource base (e.g. recycled pulp for the paper industry), high transport costs relative to CO_2 costs, safety procedures that increase transport cost (e.g. industrial gases), customer-specific products, integrated production processes, etc. However, Hourcade et al. [49] estimate the cost increase as a % of gross value added (GVA) for the 2 most affected sectors—cement and basic iron & steel—to be +34% and +27% respectively. So according to Hourcade et al. [49], anti-leakage measures should concentrate only on a few sectors, essentially cement and iron [see also [51].

Manders and Veenendaal [52] simulate the European 20/20/20 target, using the Worldscan Computable General Equilibrium (CGE) Model. Supposing negligible commitments in other Annex I countries, full auctioning and 1/3 of abatement effort via CDM, they find that ETS production would fall by 1.7% and ETS employment by 1.2% compared to baseline. They find an endogenous carbon price of €27/t CO_2 and a carbon leakage for all sectors of 0.3%. "In general we find only modest effects on competitiveness. Even in energy-intensive sectors, energy expenditure is only a fraction of total production costs. Trade flows are hardly affected, because intra-European trade is much more important than inter-European trade. Also the energy-intensive

sector is small compared to other sectors" [[52], p.33]. Note that the cost-pass through, which is an important parameter for assessing terms of trade effects, is generally not reported in CGE models. The supply curve in these models is shifted upwards including the full cost of carbon, but this creates a lower demand hence a lower equilibrium price and a cost pass-through below 100%. Since long term supply curves are flat and demand curves for goods such as steel and cement are steep, the cost pass-through will be close to 100%.

Bernard and Vielle [53], using the Gemini-E3 general equilibirum model, estimate an overall leakage rate of 7%. Terms of trade leakage of all sectors except fossil fuel exploitation is estimated at 2.3%. However, 70% of this leakage is compensated by the reduced emissions from coal mining, oil drilling and gas exploitation outside Europe [53], Table 10]

Kuik and Hofkes [42] model a carbon price of €20/t for all sectors, using the comparative-static CGE model GTAP-E. They take into account terms of trade leakage (8,8%) as well as fuel channel leakage (2.0%), both summing up to an economy-wide total leakage rate of 10.8%. They do not model technology impacts, which would lower the leakage rate. They also model the impact of border adjustment measures on total carbon leakage and conclude that their effect would be small. Total leakage would be 10.2% in the case of a common border adjustment for all foreign countries and 8.2% in the case of a border adjustment corresponding to carbon intensity in each foreign country. However, for the steel and mineral sector border adjustments would have a significant impact. For the steel sector carbon leakage would decrease from 35% without adjustment to 29% in the case of a common border adjustment and 2% in the case of a country-specific border adjustment.

The CGE model worked out by, Böhringer et al. [40] finds a very high leakage rate of 37% mainly explained by fuel-channel leakage. They estimate a 5.5% production loss compared to BAU for energy-intensive sectors, which is high compared to other estimates. Note that they obtain a high carbon price of $59/tonne for a 20% emission reduction. Their main conclusion, however, is that carbon leakage or trade effects would be affected only slightly if carbon prices were to be differentiated across sectors to minimise leakage and/or terms of trade.

FitzGerald et al. [54] find the following ranking of sectors from high to low dependence on international prices: basic metals, paper and paper products, chemicals, wood and wood products, food, beverages, tobacco and non-metallic mineral products. Combining this dependence on international prices with 2 other factors of resilience towards a price on carbon –energy expenditure shares and technical efficiency scope – they conclude that basic metals are the most vulnerable to carbon leakage.

3.4.2. Steel

Demailly and Quirion [55] investigated competitiveness in the iron and steel industry, which is one of the most exposed since it is both highly CO_2-intensive and relatively open to international trade. Assuming a 75% pass-through rate of the (opportunity) costs of emissions for the EU market and a 50% pass-through rate on the international market for a carbon price of €20/t CO_2, they find that 0% free allocation entails a significant loss of EBITDA. 57% free allocation would keep EBITDA constant (windfall profits from opportunity cost pass-through compensate production loss) and 95% free allocation would lead to 2% EBITDA increase. Varying assumptions about marginal abatement costs, price elasticity of demand and trade, cost pass-through and updating rules of allocation, they conclude that production loss compared to 2005 is very likely (> 90%) to stay under 2% and EBITDA is likely (> 60%) to increase. Carbon leakage is very likely (> 90%) to be under 15%.

Investigating basic oxygen furnace steel (carbon-intensive production process) on a time horizon of 10 years, under the assumption of a fully auctioned carbon price of €30/t CO_2, an exogenous Armington trade elasticity of 2.75 and an exogenous 50% cost pass-through, Hourcade et al. [49] find a slight increase in the non-EU imports ratio from 17% under BAU to 18%. The EBIT margin decreases from 15% under BAU to 10%. The EBIT margin goes up to 20% if allowances are 100% for free. Discussing different trade barriers, they conclude that the international pressure faced by EU steel producers is low and even if one focuses on flat products produced through blast furnaces, the pressure remains moderate. [[49], p. 92]

Kuik and Hofkes [42] model full auctioning at €20/t and a higher cost pass-through, and estimate trade effects and carbon leakage (37%) to be of greater concern compared to Hourcade et al.

3.4.3. Cement

For a fully auctioned carbon price of €30/t CO_2, an exogenous Armington trade elasticity of 1.6 and an exogenous 50% cost pass-through, Hourcade et al. [49] find for cement a short-term non-EU import ratio increase from 8% under BAU to 11% and an EBIT margin decrease from 15% to −4%. A higher cost pass-through leads to a higher import increase and a lower EBIT decrease. For 100% free allowances, EBIT increases from 15% to 38% [see also [56]]. For both steel and cement they conclude that "Short-term market share or profit margin losses may be very high under unlikely sets of assumptions on pass-through ability, trade sensitivity, allocation methodology and CO_2 price. This conclusion would be challenged if international pressures on EU producers increased—this hypothesis is highlighted by industry but quantitative evidence gives support to the contrary [[49], p. 93]."

In line with the findings of Hourcade et al. [49], FitzGerald et al. [54] estimate the vulnerability of the non-metallic mineral products (essentially cement) to be intermediate to low because of their low exposure to international competition and intermediate technical efficiency scope.

Ponssard and Walker [57] find a high import increase from 10% in the baseline case to 20% in the case of a fully auctioned carbon price of €20/t CO_2, leading to a carbon leakage of 70%. Previous studies [58,59] found lower carbon leakage values of respectively 29% and 30%. Ponssard and Walker's high value may be explained by the assumptions of a relatively high cost pass-through of 67% and no (short-time) abatement opportunities.

3.4.4. Chemical industry

Tomás et al. [60] assessed the impact of the ETS on the direct and indirect production costs of 4 representative Portuguese production units of the chemical industry. The chemical industry is not integrated as such in the EU ETS, but all combustion installations above 20 MW capacity are and the chemical industry is itself a large electricity consumer. Tomás and co-authors find marginal cost increases (at a fairly low price of €15/t CO_2) between 0.60% and 2.26%.

3.4.5. Refinery

Reinaud [61] finds that a fully auctioned carbon price of €10/t would incur 15% to 30% extra costs to refineries (as a % of refinery margin). This is a high-end estimate because abatement opportunities are not taken into account. For certain products – aviation gasoline and kerosene, for example – several European countries are almost exclusively supplied by European refineries, which could pass through carbon costs. Most refined products, however, are subject to international competition and thus a fully auctioned carbon price would significantly decrease profit margins. Nevertheless, several trade barriers such as very high European desulphurisation norms

and recent under-capacity in high growth regions temper international competition. Smale et al. [56] find a much lower cost increase.

3.4.6. Aluminium

Smale et al. [56] found that indirect costs from €15/t CO_2 render the UK aluminium sector completely uncompetitive, causing a full loss of its market share but this contrasts with the ex-post studies discussed hereafter.

3.4.7. Aviation

Anger [62] models the effect of the inclusion of the aviation sector in the EU ETS in accordance with the EU directive [6] for the 2012–2020 period. At a carbon price of 20€/t, she finds a negligible effect on demand, but by 2020, aviation emissions are reduced by 3.4% as a result of a lower carbon intensity of the sector. GDP is not affected because revenues induced by the ETS are recycled in the economy. However, it is estimated that the emissions induced by this revenue recycling offset the reduction in emissions in the aviation sector.

3.4.8. Ex post analysis

Different *ex post* studies of the first period do not reveal a significant effect of ETS on competitiveness.

Applying an econometric *ex post* regression analysis for 419 German ETS firms, Anger and Oberndorfer [63] find no influence of the allocation factor (allowances/emissions) on revenues or employment. Ellerman et al. [23] regressed net imports on demand and carbon price for the European cement, steel and aluminium industries. In none of these sectors did they find a significant impact of carbon price on net imports.

Reinaud [39] also concludes that the EU ETS has not, so far, triggered observable carbon leakage in steel, cement and primary aluminium sectors. Reinaud [64], using a regression analysis like Ellerman et al. [23] and a Chow test, does not find a significant effect of carbon price on net aluminium imports either. Between 1999 and 2006 "the playing field remained approximately the same around the world, and even worked in favour of Europe as a whole, as costs increased less than the global average. Nonetheless, it costs more to produce one tonne of primary aluminium in Europe than it does in many competing countries" [[64], p. 30]. Basing his study on a Chow test, Lacombe [65] does not find a significant impact on net imports in the refinery industry.

Barker et al. [45], using the Computable General Equilibrium model E3ME, investigate *ex-post* the carbon leakage from six EU Member States that implemented environmental tax reforms unilaterally over the period 1995–2005. They model a BAU counterfactual taking into account terms of trade, technology development induced by higher energy costs and revenue recycling. Note that the environmental tax level is moderate (revenues maximum 1% of GDP) and that the fuel price leakage channel is underestimated because extra-European countries are not in the model. They find that the tax reforms did not increase, but slightly diminished emissions abroad (i.e. negative carbon leakage) because of technology spill-overs.

3.4.9. Conclusion on leakage

Leakage, which is closely linked to long-term investment decisions, is difficult to detect over the short time frame of the first three years of the ETS. Leakage patterns may be different in the future as the first phase was characterised by (1) over-allocation for industrial sectors, (2) increasing steel, aluminium, cement and petroleum world prices and (3) an economic upturn. Reinaud [39] stresses the importance of a continuous analysis of cost pass-through capacity, trade flows and investment decisions as indicators of carbon leakage. Understanding how much competitiveness is lost is a prerequisite as current anti-leakage measures are costly (see further).

Ex ante model estimates of carbon leakage vary considerably. But most conclude that even if a significant proportion of allowances is auctioned, competitiveness effects are fairly low. Given the fact that industry was on average 100% grandfathered, the ETS increased the mean profitability of ETS plants.

4. Economic efficiency

Mickwitz [8] distinguishes economic efficiency in the sense of cost-benefit (are the benefits worth the costs?) from economic efficiency in the sense of cost-effectiveness (could the results have been achieved with fewer resources?) [see also [4]]. The cost-benefit relation, which is the most comprehensive criterion, will be evaluated first and the cost-effectiveness, which is a more measurable and more usable criterion, will be discussed afterwards. Next, transaction costs are discussed and the effect on technology development is the last aspect that will be covered under the evaluation of economic efficiency.

4.1. Cost-efficiency (cost-benefit analysis)

The marginal Social Cost of Carbon (SCC) is defined as the actualised cost of all climate costs (and benefits) in the future caused by emitting one extra tonne of CO_2 today. As the CO_2 concentration in the atmosphere rises in the future, marginal damage will increase and the SCC will rise at an estimated rate of 2–3% per year [66]. The SCC can be increased to correct for risk aversion, discontinuous and irreversible climate damage, inequality aversion, and so forth.

Many studies have estimated the SCC with great differences in outcome. A first difficulty is that the SCC depends on estimated future emissions. Optimistic assumptions about future abatement costs reduce present-day SCC and vice versa [67]. An even more controversial debate concerns the discount rate. Most studies apply a Ramsey discount rate $\delta = \rho + \eta g$ with ρ the pure time preference rate, η the elasticity of marginal utility with respect to consumption and g the growth rate of consumption. The most polemic aspect is the pure time preference rate – the rate at which the utility of future generations is discounted just because they are in the future – which is seen as a strictly ethical judgement [66].

Tol [68] developed a Joint Probability Density function of the SCC based upon 211 estimates of the SCC from 47 studies between 1982 and 2006. He obtains a mean SCC (in 1995 $) of:

- $35/t CO_2 for all 211 estimates,
- $86/t CO_2 for the estimates with a pure time preference rate of 0%,
- $19/t CO_2 for the estimates from peer-reviewed literature
- and $24/t CO_2 for estimates after 2001.

One of the most debated and voluminous studies was made by Stern [67], who estimated a high SCC of $85/t CO_2. The fact that he applied a pure time preference rate of 0.1%, resulting in a discount rate of 1.3%, was subject to vocal academic debate. On the low side of the cost spectrum, Nordhaus [69] estimates the SCC in the case of optimal climate policy at only $7.4 per tonne CO_2 (in 2005 $). Note that this estimate is based on a discount rate of 4% (opportunity cost of capital) and a welfare maximising global warming of 3.4 °C in the 22nd century.

Aldy et al. [66] discuss estimates from 2006 on and conclude that the 450 ppm CO_2-eq target implies emissions prices corresponding to an SCC of $40–90 per tonne CO_2 by 2025, while the 550 ppm CO_2-eq target implies an SCC of $3–25 by 2025 (in 2000

$). These targets are consistent with a best estimate for mean global temperature increases of 2.1 °C and 3.0 °C, respectively [14]. So if global warming is to be limited to less than 2 °C as stated in the Copenhagen agreement, the former estimates of $40–90 per tonne constitute a minimum.

Besides the discount factor, a second intensive debate surrounding the social cost of carbon is the treatment of catastrophic risk. Weitzman [70] shows that if the structural uncertainty surrounding climate change induces a critical "tail fattening" of the probability density function of climate damage, these extreme risks may outweigh the effects of discounting in climate policy analysis. The certainty-equivalent marginal damage from current emissions then becomes infinite. On the other hand, advocates of low-risk premiums argue that, if a climate catastrophe appears likely to happen in the future, mid-course corrections of abatement efforts should be taken into account when evaluating risks today.

The Marginal Abatement Cost (MAC) is the cost to mitigate one extra tonne of CO_2. The abatement path that maximises wealth realises all abatement opportunities for which MAC < SCC. The MAC can be decreased to account for dynamic effects: technology developed from abating today makes future abatement cheaper. The MAC also depends on restrictions on other forms of pollution. Pietrapertosa et al. [71] find for a local energy system a MAC that is twice as big when soot is to be limited to the BAU emission.

When markets are functioning perfectly, creating carbon permits or setting a carbon tax will trigger abatement opportunities that are cheaper than the tax or the carbon price. As the cost of the externality (climate damage) is internalised in the price of polluting goods, demand will shift to less polluting substitutes and contribute to cost-efficiency.

As the majority of estimates of the SCC are much higher than the EU ETS carbon price, the cap of the system was probably too unambitious to be cost-efficient. The actual price decrease below €8/t in 2012, induced by the economic crisis, has made the low abatement target a major weakness of the present EU ETS.

4.2. Cost-effectiveness

Given a particular environmental goal (the Kyoto target in the case of the EU ETS), the most cost-effective policy is the one which achieves the desired goal at the least cost [14], see also [7]. The cost-effectiveness of the EU ETS was a central argument when the European Commission proposed the ETS [1,72].

Since marginal abatement costs increase with abatement, equalising them over different plants is the cheapest way to attain a given abatement target. This basic argument is straightforward and is acknowledged in all academic studies. So cost-effectiveness in general is a strong point of a cap-and-trade system, even if there are various obstacles to cost-effectiveness.

Since the EU ETS was set up to realise the Kyoto targets of the European countries, the attribution of allocations with the lowest overall cost is obtained when the marginal abatement costs for companies inside the ETS equals the marginal abatement cost for companies outside the ETS. Böhringer and Rosendahl [73] test if countries that are very large net sellers (or buyers) of quotas have an advantage of exploiting their market power and allocate less (or more) than the above optimum in order to increase (or decrease) the carbon price. They conclude that single countries can significantly affect the outcome of the EU ETS by exploiting their market power, leading to quite differentiated marginal abatement costs outside the EU ETS, but that the effect on the quota price is small. There is a consensus about differentiated marginal abatement costs in the non-ETS sectors, confirmed by computable general equilibrium studies such as [53] and [74].

Under the assumption of efficient markets, carbon prices are driven by the marginal abatement costs of the participating firms. However, Hintermann [75] finds that before the price crash in April 2006, carbon prices were most likely not driven by market fundamentals determining marginal abatement costs (fuel prices, temperature, availability of hydroelectric power). Prices may have been affected by self-fulfilling expectations (a bubble), by firms with market power or by firms hedging against stochastic emissions, but these hypotheses are difficult to test. The carbon price most likely did not equal the marginal abatement costs, which decreased cost-effectiveness. After April 2006, carbon prices were found to be much more correlated to market fundamentals.

Free allocation has different distortional effects and lowers cost-effectiveness. Next, distortional incentives of updating, new entrants and closures will be assessed. Windfall profits, also induced by free allocation, are discussed in the chapter on distributional effects.

4.2.1. Distortional incentives of updating: free allocations based on emissions or production

The first time free allocations are distributed, they can be considered as lump-sum transfers without distortional incentives. But to grandfather the same yearly amount into the infinite future is impossible for reasons of equity between companies. So at a given moment free allocations will be based on former emissions or production levels.

Let us first consider free allocation based on past emissions, without a price cap, without making links to other emission trading systems, and without banking nor borrowing. Let σ_t be the carbon price in period t and δ the discount rate. If a proportion λ_{t+k} of emissions in period t is freely allocated in period $t+k$, then the first-order condition with respect to emissions of the profit-maximising firm is [76], based on [77]:

$$\sigma_t - \frac{1}{(1+\delta)^k}\lambda_{t+k}\sigma_{t+k} = MAC_t \tag{4}$$

For an auctioned allocation, the first-order condition with respect to emissions of the profit maximising firm gives

$$\sigma_t = MAC_t \tag{5}$$

Compared to auctioning, the second left-hand term of Eq. [4] accounts for the revenue loss of a fraction λ of future free allowances at time $t+k$, because of abating one extra tonne at time t. As the left-hand side of the equation is equal across firms, marginal abatement is uniform among firms, and unaffected compared to full auctioning since the cap is the same, resulting in the first-best cost-efficient solution. However, the carbon price will be much higher than the marginal abatement cost. If banking and borrowing of allowances are allowed, (Eq. [4]) simplifies to [76]:

$$(1-\lambda_{t+k})\sigma_t = MAC_t \tag{6}$$

So for an allocation rate of 95%, the quota price will be 20 times higher than MAC. However, this only holds for a closed market without a price cap or without linking to an external market. The EU ETS is linked with the flexible Kyoto mechanisms CDM and JI. Linking with these external markets has the effect of a price cap. In this case, emission-based updating strongly reduces abatement efforts and results in a sub-optimal solution.

Most Member States allocated free allowances to existing installations taking into account emissions of base periods extending to 2005, rewarding firms that postponed abatement in order to receive more free allowances in the 2nd period. This violation of the guidance provided by the Commission may be explained by the fact that no verified emission data existed before 2005. Anderson and Di Maria [25] indeed observe "inflated"

emissions (emissions above business-as-usual) during the first phase for certain countries.

The phenomenon of postponed abatement in order to obtain more free allowances may have occurred already before the 1st phase of the ETS. Arto et al. [78] show that early action to reduce emissions through technology and energy mix between 1995 and 2005 was not (fully) accounted for in National Allocation Plans (NAPs), rewarding companies that postponed abatement.

Free allocations can also be based on previous production levels. This type of free allocation is often called "baseline and trade" or "benchmarking". With this type of updating, free allowances equal the emission rates of the best available technologies in the sector (or are a proportion of this benchmark). This will be the EU ETS policy starting in 2013. Under output-based free allocation, abatement decisions are optimal, corresponding to Eq. (5), but quantities are affected. Let λ_{t+k} be the (benchmarked) free allocations per unit of output. The first-order condition of the profit-maximising firm with respect to quantity becomes [76,77]

$$p_t = \textit{Marginal (low carbon) Production Costs}_t - \frac{1}{(1+\delta)^k}\lambda_{t+k}\sigma_{t+k} \tag{7}$$

These output-based free allocations can be seen as an output subsidy that is proportional to carbon intensity, leading to lower prices for carbon-intensive goods, in comparison with prices under full auctioning. This means that output-based allocation rules prevent cost pass-through. Hence windfall profits are strongly reduced (see further).

In an open trading system, the first-best solution requires auctioning. Nevertheless, according to Böhringer and Lange [77] second-best rules correspond to a Ramsey formula and are generally based on a combination of output and emission levels. This corresponds to an optimal arbitrage between carbon price distortions under emission-based allocation and product price distortions under output-based allocation. If limited consumption-induced abatement due to no cost pass-through is judged politically more desirable than carbon leakage, then output-based allowances are the best option.

4.2.2. Distortional incentives for new entrant endowments and closures

All Member States foresee new entrants' reserves. So new installations receive free allowances, based on installed new capacity. This applies to new companies as well as to expanding existing companies. Only power plants in the Swedish power sector have to buy all their allowances on the market during the second phase [4]. Most Member States also decided to end the distribution of allowances with the year an installation closes [4].

Ellerman [79] derives the first-order conditions of profit-maximising firms under new entrant endowments and closure withdrawals. A new entrant allowance endowment lowers the entry price condition for determining investment in output capacity. The expected value of the new entrant endowment acts as a lump-sum payment that helps to offset the annual capital recovery charge. Consequently, the effect of a new entrant endowment is to increase output capacity. New entrants' allowances induce a production capacity subsidy-effect on polluting industries (also [4,11]). Competition between Member States to attract new plants creates an extra incentive to grandfather even more generously to new plants than to established installations.

The introduction of a cost for emissions (opportunity cost under free allocation or real cost under auctioning) would lead agents to close more facilities than they would if this cost element were not present. But in most NAPs, closures lead to termination of free allocations. The value of this lost free allocation creates an incentive to keep old installations running. This leads to fewer closures than would be the case when closed installations continue to receive free allocations after closure. In the case where free allocation exceeds emissions (old installations are often long because production is below capacity), closure provisions lead to even fewer (later) closures than would be the case without carbon constraints [79].

So both new entrants and closure provisions create overcapacity, resulting in higher output and lower prices (depending on demand elasticity) than would be the case under an ETS without these provisions.

4.3. Transaction costs

Transaction costs are composed of administrative costs (by governments) and compliance costs (by the ETS companies). Firms' compliance costs include:

- early implementation or start-up costs (familiarisation with rules and guidelines, calculation of baseline emissions, measuring equipment, consultancy),
- monitoring, reporting and verification (MRV) costs (annual emission report, hiring an accredited emission verifier, consultancy services),
- and trading costs (access to carbon exchange or brokerage fees) [80].

Trading costs also include trading gains made by intermediary sellers and buyers, since these gains create a gap between the income of long installations and the expenses of short installations [35,36].

Based on a survey of 27 Irish firms, [80] find mean early implementation costs to be €72 thousand per firm and mean MRV costs to be €74 thousand over the first 3-year period. These MRV costs are unevenly distributed among installations of different sizes. Expressed as cost per tonne of CO_2 emission, large, medium and small companies incurred MRV costs of respectively €0.02/t, €0.56/t and €1.51/t. The cost for large emitters is reasonable, but the cost for small emitters is problematic. Surveying 114 Swedish companies, Sandoff and Schaad [29] find a mean time investment of 27 man hours per month for ETS compliance. This time investment varies from 17 to 18 to 42 h for small, medium and large companies, respectively. 40% of incumbents, mostly small and medium-sized companies, stated that the transaction burden on firms with low emissions is too high and not proportional to their environmental impact. For most companies, however, Sandoff and Schaad conclude that this time investment is fairly moderate.

Administrative costs from the government add to the compliance costs of firms. Betz and Schleich [81,82] estimate start-up costs of the ETS in Germany at €7.5 million and recurrent costs at €7 million per year. Wittneben [36] also mentions the cost of legal actions from companies against allocation decisions made by their governments. Several countries also contested in court the downward revision of allocations in their NAPs by the Commission.

Carbon taxes have lower transaction costs as they are simpler and more transparent to apply and their application can use the existing tax structure [39,80,83,84].

4.4. Dynamic cost-efficiency: innovation

Technological change is difficult to model, but has a tremendous impact and is the major contributor for dynamic cost efficiency. Neij and Astrand [44] state that most impact evaluations are focused on the direct results of policy implementation and lack information on how policy instruments did, or did not, affect the process of technological change.

4.4.1. *Price uncertainty of ETS dampens incentive for innovation and investment*

The value of future emission reductions under ETS creates a market incentive for innovation in low-carbon technologies. Uncertainty on carbon prices, however, makes the returns of low-carbon investments more variable and therefore increases the cost of capital (the minimum expected return to capital demanded by investors) of these projects, thus shrinking investment opportunities.

Blanco and Rodriguez [85] illustrated this in the wind turbine sector. They found that the 6 countries that had installed more than 250 MW wind power in the EU in 2006 had a feed-in tariff (or equivalent regulation) that was equivalent to a carbon price of €40.6/t CO_2 in Germany, €45.5/t CO_2 in Spain, €33.3/t CO_2 in France, €31.3/t CO_2 in Portugal, €24.47/t CO_2 in the UK and €159.4/t CO_2 in Italy (in decreasing order of investment). The relatively low prices for the first 4 countries are explained by guaranteed support for 10–25 years. According to Blanco and Rodriguez [85], price uncertainty explains why Italy had by far the highest feed-in tariff and a mitigated investment level.

The negative effect of price uncertainty on long-term investments was also observed in the conventional electricity sector. Conducting interviews among the German electricity producers (covering 80% of the sector's emissions), Hoffmann [86] concludes that the first phase of the EU ETS constituted a main driver for small-scale investments with a quick payback time of typically 1–5 years. But the impact of the EU ETS on large-scale investments in power plants or in R&D efforts is found to be limited, essentially because of regulatory uncertainty. Researching the response of the Finland energy sector during the first phase of the ETS, Kara et al. [87, p. 202] conclude that in Finland, market actors seemed to delay investment decisions in emission reductions rather than to realize them during the first phase of the emission trading system. Based on an agent-based model (simulating different management styles) for the Dutch electricity sector in the coming decades, Chappin and Dijkema [89] predict that the effect of ETS on emissions, with an endogenous carbon price between €10/t and €50/t, is relatively small and materializes late. Conventional coal would still remain an important option. Indeed, half of the projected investment in new electricity capacity in Germany and the Netherlands concerns coal plants [89]. For the entire European electricity sector, Ellerman et al. [23] cite that planned investment in new coal electricity capacity has not diminished since the EU ETS (27% of new capacity in 2007, based on Platt's Powervision Database). This new coal capacity will engage coal-fired electricity for several decades, even if carbon prices seriously increase in the future.

However, carbon price uncertainty may have a differentiated effect in different companies. Based on qualitative research, Sandoff and Schaad [29] describe 2 opposite investment strategies induced by carbon price uncertainty. The first strategy, as described above, sees abatement as a way to maximize expected profits, considers risks to be calculable and is confident in the allowance market as an alternative to abatement. As risk pushes up the cost of capital of abatement projects, abatement investments decrease with greater uncertainty. Under this second strategy the company sees uncertainty as highly disturbing and investments are made because managers are willing to take on costs for decreasing exposure to this uncertainty. This strategy puts considerable focus on the internal alternatives. In this case, higher uncertainty leads to higher investment. In their survey of 114 Swedish ETS companies, they find that internal reduction of CO_2 is seen as the most important measure for handling a potential deficit.

Neuhoff [51] distinguishes a similar duality in assessing the effect of carbon price uncertainty on investments, differentiating companies' long-term and short-term targets. Concerning short-term targets, investors in individual projects typically focus on the return over the initial 5–10 years. They often consider that carbon prices and volatility at the time of investment are representative of future prices. For such investors, stable and predictable carbon prices reduce uncertainties and risk, thereby facilitating access to cheaper capital. Oil and technology companies have long-term perspectives on the market share of fuels, technologies and their products. They need a credible long-term policy framework, where the carbon price is likely to rise, to ensure an appropriate market share for low-carbon technologies. In this case, the price-increasing assumption under ETS stimulates more innovation than a stable carbon price would (under a carbon tax for example).

4.4.2. *Focus on intermediate technologies, not enough on 'high-abatement' technologies*

Abadie and Chamorro [90] find that investing in a carbon capture & storage facility is profitable only if allowance prices exceed €55. Von Stechow et al. [15] argue that if a carbon market was chosen as policy to trigger carbon capture and storage, a (high) carbon price guarantee would be needed to decrease investment uncertainty. Blanco and Rodriguez [85] (see above) found that of the 6 countries that had installed more than 250 MW wind power in the EU in 2006, all had a feed-in tariff (or equivalent regulation) that was equivalent to a carbon price between €25 and €159/t CO_2. Photovoltaic energy, tidal energy, geothermal energy and hydrogen would in most cases require even higher carbon prices to be developed by a price on carbon as single incentive.

R&D in these technologies today is important because it decreases the cost of long-term high levels of abatement (-80% to -95% in 2050 or later). Intermediate technologies (such as combined cycle gas turbines and coal efficiency) make the mid-term abatement cheaper, but do not decrease the cost of high levels of abatement, because intermediate technologies need to be substituted away from to attain higher levels of abatement. As intermediate technologies do not affect the long-term cost, they decrease MAC in the short run and increase MAC in the long run [[91], p. 48]. The low incentive for high-abatement technology is one of the reasons why Capros et al. [74] argue for a renewable energy target besides the EU ETS. The inclusion of an explicit RES target is important to enable learning-by-doing and economy-to-scale effects to take place dynamically.

A cap-and-trade regime is designed to concentrate emission-reduction efforts on those facilities that have the lowest abatement costs. The in-built orientation on lowest-cost emission mitigation orients abatement effort to the low-end technology spectrum and fails to account for the syndrome of falling abatement costs—that is, the propensity of initially expensive innovations to develop in the longer term (notably through economies of scale) into low-cost abatement mechanisms [[11], p. 199]. Pricing carbon will increase the market value of high-end abatement technology. If a technology turns out to be competitive once it has been wideely deployed, the carbon credits induced by this technology increase its profitability, creating an incentive for high-end R&D today. But uncertainty about technology potential, uncertainty of future carbon prices and a high discount rate (cost of risky capital) reduces the incentive tremendously. Moreover, innovations, even if they are patented, can be partially copied by competitors. This knowledge spillover reduces the appropriability of the return of R&D investments, also leading to sub-optimal R&D. Fischer and Newell [92] find that a carbon price (ETS of carbon tax) combined with R&D subsidies that compensate for

knowledge spillovers are significantly more efficient than a single policy of emission trading.

According to the literature on socio-technological regimes, a carbon market or carbon tax creates room for low-carbon niches by making the 'gradients for action' more favourable to low-carbon socio-technical practices. However, technology development is considered to be highly uncertain, constrained by routines, and therefore difficult to boost through carbon prices. As such, a price signal will be insufficient to trigger the uncertain process of transforming technological regimes [93]. A price on carbon will be insufficient to overcome lock-in of fossil-fuel-based technology systems [94] and highly centralised large-scale electricity generation [95,96].

5. Distributional effects

5.1. Windfall profits

To the extent that prices are driven by short term marginal costs, producers pass through the opportunity costs of emissions trading to sales prices even if they receive allowances for free. This creates windfall profits. The cost pass-through of opportunity costs is especially high for sectors without extra-European competition, like power generation. In the power sector, windfall profits are not only induced by free allocation. The price increase created by high-carbon, price-setting technologies induces windfall profits for lower-carbon, non-price-setting technologies, such as nuclear energy. The former windfall profits disappear with auctioning, but the latter do not. This specific price-setting in the electricity sector explains why all windfall profit estimates exceed the market value of allowances. Note that the latter effect creates a competitive advantage for carbon-sober electricity which is part of the intended effect of the ETS. Extra profits induced by the ETS may also be due to monopoly strategies of big electricity companies [97].

Sijm et al. [98] assess the impact of EU ETS on electricity prices in a report to the Directorate-General for the Environment of the European Commission. They find *ex post* regression estimated cost pass-through rates on the forward markets in 2005 and 2006 in five countries (France, Germany, the Netherlands, Sweden, the UK) between 38% and 83% in most cases, with a minority between 103% and 134% and an outlier of 182%. Cost pass-through rates on spot power markets were less significant because of many other important price-determining factors. For a €20/t CO_2 price, based on scenario model results, they find EU-20 power windfall profits of €28.0 billion under perfect competition and €24.4 billion under oligopolistic competition.

Lise et al. [99] estimated windfall profits with the COMPETES model. Electricity producers from 20 European countries are modelled under 11 scenarios with perfect competition vs. oligopolistic competition—demand elasticity varying between 0 and 0.2 and an exogenous carbon price of €20 and €40/t. They find mean price increases of €10–13/MWh, most cost pass-through rates between 70% and 90% and windfall profits between €24 and €35 billion for a €20/t carbon.

Keppler and Cruciani [100] as well as Kara et al. [87] model the electricity market assuming a full opportunity cost pass-through. Keppler and Cruciani, based on a mean carbon price of €12 during the first ETS period, find windfall profits of €19 billion per year. They also find €10 billion windfall profits under the 3rd period when allowances will be fully auctioned (supposing a carbon price of €20/t). This is explained by the fact that infra-marginal low-carbon producers continue to have rents when high-carbon producers are the price-setting technology.

Based on a carbon price of €10/t CO_2, Kara et al. [87] find an *ex ante* windfall profit of €340–450 million/year for electricity production in Finland, depending on the amount of free initial allocation of emission allowances. Taking into account the lower carbon price, this value is comparable to the findings of Sijm et al. [98]. Given that the 2 first studies [98,99] model CO_2 prices above the mean price of the first period, it can be concluded that windfall profits for the first phase are estimated between €19 and €25 billion per year.

Other authors did not estimate windfall profits but did calculate cost pass-through, which is an indirect measure for windfall profits as the great majority of the emissions were covered by free allowances. Fell [101] conducted a cointegrated vector autoregressive (CVAR) model of the link between allowance prices and electricity prices in the Nordic electricity market (Sweden, Norway, Finland and Denmark). Their impulse response analysis indicates short-term price responses that are coherent with full cost pass-through for coal, even if coal generation constitutes only 6% of the market. The long-term cost pass-through response, however, is lower. Cost pass-through in other sectors than electricity was also detected [49,56], but of a lower magnitude because of international competition. The results are summarized in Table 3.

Oberndorfer [102] as well as Veith et al. [103] find a positive correlation between price developments of EU emission allowances and stock returns for the largest European electricity corporations. This indicates that carbon allowances are seen as a source of windfall profits rather than as a cost to shareholders.

Conducting qualitative research on strategic responses to climate policy, Pinkse and Kolk [104] conclude that energy producers have been most successful in acting as institutional entrepreneurs, actively lobbying to obtain ETS design to their advantage, at the expense of electricity consumers.

5.2. Social distributional effects

Carbon taxes and auctioned emission schemes, if applied to the same sectors, induce the same price increases and therefore have a very similar effect on income distribution [105]. Since a long time, carbon taxes are found to be regressive—the burden is more than proportionally borne by poorer households [106,107].

Feng et al. [108] investigate the distributional impact of a 30% greenhouse gas reduction compared to 1990 emission levels in the

Table 3
Ex post and *ex ante* estimates of cost pass-through and windfall profits.

Author	Region	Carbon price (€/t CO_2)	Cost pass-through	Windfall profit (€billion/year)
Sijm et al. [98]	EU	20	38% to 83% in most cases, with a minority between 103 and 134% and an outlier of 182%	€28 B/year under perfect competition €24 B/year under oligopolistic competition
Lise et al. [99]	EU 20	20	70–90%	€24–35 B/year
Keppler and Cruciani [100]	EU	12	100% exogenous	€19 B/year
Kara et al. [87]	(Fi, No, Sw)	10	100% exogenous	€0.34–0.45 B/year for Fi €1.7 B/year for No and Sw

UK, realised through a carbon tax of £93/t. A CO_2 tax would cost 6% of income for the lowest income decile, and only 2.4% of income for the highest decile. If all greenhouse gases are regulated, costs represent 4.3% of the lowest incomes and 1.7% of the highest incomes. The impact of regulating all greenhous gases is less regressive because heating and electricity (CO_2 emissions) are even more overrepresented in expenditures of the poor than is food (CH_4 and N_2O emissions).

Callan et al. [109] find that a carbon tax of €20/t on home heating, motor fuels and electricity emissions in Ireland would cost the poorest households €3.8/week and the richest households €5.1/week, confirming the highly regressive effect of a carbon tax. However, the carbon tax would become neutral if all social benefits were increased by €2/week and if the basic personal tax credit were increased by €104/year. This would still leave a net income to the state.

Windfall profits under ETS make emission trading even more redistributive than under a carbon tax. Taking windfall profits of grandfathered allowances into account, Dinan and Rogers [105] find that for a 15% reduction in CO_2 emissions by an ETS, US households in the lowest income quintile would be worse off on average by around €500 per year, while households in the top income quintile would reap a net gain of around €1000. Parry [110] also concludes that grandfathering is excessively income-regressive.

As the regressive nature of carbon policies is politically sensitive, all authors propose compensating measures to distribute the tax burden progressively among the population. The revenues needed for this redistributional policy are another argument for auctioning instead of free allocation.

5.3. Intercountry and intersector transfers

Over the course of the first three years, 650 million allowances from long installations were sold to short installations. This is about 11% of the EU-wide cap, implying a market value transfer of €5.2 billion between installations. The total quantity of allowances traded across country borders (gross imports or gross exports) was 350 Mt [111]. Net imports (=net exports) were 21 Mt, 60 Mt and 141 Mt during 2005, 2006 and 2007, respectively, totalling 218 Mt and implying an estimated financial flow of €941 million. The largest exporters were Poland (53 Mt, €93M), France (42 Mt, €203M), the Czech Rep. (28 Mt, €201M) and the Netherlands (13 Mt, €59M). The largest importers were the UK (−107 Mt, -€457M), Spain (−41 Mt, -€279M), Italy (−35 Mt, -€169M) and Germany (−30 Mt, +€9M) [33]. As Eastern European countries generally had very long positions, the net import of allowances by the EU-15 coming from the EU-10 is estimated at approximately €505 million over phase I [111]. Trotignon and Ellerman [33] conclude that size and value of cross-border trading were relatively low, but that frequency and density of trade werehigh.

Concerning intersector transfers, based on 2005–2006 data, Kettner et al. [112] calculated that the 7370 smallest installations (totalling 5% of emissions) were on average long (33%) and the 179 biggest installations (totalling 50% of allocations) were on average short (−5.8%). Also, discrepancies between allowances and emissions were much more variable among small installations. Power and heat were on average short (−5%), while other sectors were on average long (11.6%). Trotignon and Delbosc [111] found a comparable net short position for electricity producers (−6.1%), leaving all other sectors with a net long position (ranging from 4% for cement to 20% for paper and board).

Even if intercountry and intersector transfers are on average reasonable, it can be concluded that the distributional effects from low income to high income families have constituted a major drawback of the ETS. There is an academic consensus on windfall profits and its regressive effect on income distribution. Correcting for these distributional effects will be one of the challenges in the future if the ETS is to gain durable political support. In this respect, full auctioning for the electricity sector from 2013 is a step in the right direction.

6. Institutional feasibility

According to the IPCC [[4], p. 752] Policy choices must be both acceptable to a wide range of stakeholders and supported by institutions, notably the legal system. Other important considerations include human capital and infrastructure as well as the dominant culture and traditions. The decision-making style of each nation is therefore a function of its unique political heritage. Emission trading was not part of the European political heritage. Why did the ETS – unlike the European carbon tax – passthrough the European decision-making process? This will be the first question to be discussed. Next, the reliability of national registries and the complexity of the ETS will be addressed. Finally, the ambiguous role of free allocation will be handled.

6.1. How the ETS gained support among the European Commission, industry and a number of NGOs

In 1992, the European Commission proposed an EU-wide carbon tax. After intense lobbying from industry and the reluctance of certain Member States, the Commission abandoned the project. During the Kyoto negotiations the European Commission firmly opposed carbon trading. Only in the ultimate phase of negotiations were flexible mechanisms of emission trading included in the protocol on the insistence of the US delegation under the leadership of Al Gore. Braun [113] points out 3 major preconditions that explain the evolution of the European Commission from emission-trading opponent in 1997 to architect of the largest carbon market in the world a few years later:

- The unsuccessful attempt to introduce an EU-wide carbon tax. Unlike a carbon tax that needs a unanimous vote, emission trading only requires a qualified majority vote in the European Council.
- The failure of the 6th Conference of the Parties to the UNFCCC in 2000 and the withdrawal of the US from the Kyoto Protocol. The Protocol would only be implemented if at least 55% of the UNFCCC signing countries with at least 55% of the emissions of industrialized (annex I) countries ratified the protocol. So the protocol risked not being implemented at all. Voting the European cap-and-trade system in 2003, when it was not clear that the Kyoto Protocol would be implemented, was a clear sign to other countries that Europe was serious in its ambition to reach its emission target.
- The inclusion of international emission trading in the Kyoto Protocol.

Falkner [114] focuses on the role of divergent interests among industry. He describes how major industrial companies moved from a monolithic anti-climate-regulation block behind the US-based "Global Climate Coalition" to more diverse positions relating varying interests of different industry branches. A growing faction deemed that climate warming could be a threat (e.g. the insurance industry) or create business opportunities. In 1999, BP created its own company-based emissions trading system and achieved a 10% emission reduction between 1990 and 2002. Shell, the cement producer Lafarge and UNICE (now the employers' confederation BusinessEurope) also counted among early supporters of emission trading [113,115]. In the beginning of 2000, most industries felt that some kind of regulation was becoming inevitable

and lobbied for emission trading, supposing that, at least for a long initial phase, allowances could be obtained for free [38].

Molina and Guerrero [116] point out that the high potential climate damage and public awareness favoured Europe becoming the first mover in climate policy. They estimated that willingness to pay for climate damage in 1995 was 6 times higher in European OECD countries than in non-European OECD countries.

As there was no experience with emission trading in Europe, the Directorate-General for the Environment (DG Environment) of the Commission built up knowledge by contracting the external consultants Foundation for International Environmental Law and Development and the Center for Clean Air Policy in Washington, DC. This led to the Green Paper on emissions trading in March 2000 [1]. From March 2000 to January 2001, the DG Environment managed to convince representatives from industry and environmental NGOs to back emission trading, through discussions in the European Climate Change Programme (ECCP) Working Group 1 on flexibility mechanisms. To industry, the instrument was framed as a cost-effective tool that could even provide economic opportunities for shrinking emitters to sell allowances. To environmental NGOs (and the European Parliament) the instrument was framed as environmentally effective, as it would automatically lead to the target [[72], p. 319]. To national governments the above arguments were combined, adding that emission trading was effective for reaching their Kyoto targets. The DG Environment of the European Commission, managing know-how from consultants, business and NGOs, had the possibility of always being a step ahead as regards knowledge on emission trading. By doing so, the DG Environment acted as a policy entrepreneur in the policy network of the European Union [72,113,117].

Initially, only the UK, Denmark, the Netherlands, Sweden and Ireland supported emission trading [72]. None of them, however, took the initiative to push the idea of a European ETS. Yet by 2001 a majority of the Member States could agree on the basic ETS design and on 13 October 2003, Directive 2003/87/EC was adopted by the European Council and the European Parliament [2].

6.2. Technical complexity of trading

Any cap-and-trade system depends on the registration, audit, enforcement and control activities being ensured over time by a group of people who are committed to a transparent, secure and fair market, because it concerns an intangible market with artificial scarcity. Thus the EU ETS is institutionally demanding because in case of large-scale fraud, the perennity of the market is at stake. This constitutes a disadvantage over a carbon tax [35,36]. In 2009, high trading volumes on Bluenext were provoked by a value added tax (VAT) carousel. In February 2010, a phishing attack allowed criminals to obtain the access codes of 6 German companies, enabling then to sell 250,000 allocations for more than 3 million euros. And in January 2011, another attack led to the theft of 28 million allocations from 5 national registries, leading to the closure of all EU ETS registries for several days [118].

Conceptual complexity, making the policy more difficult for citizens and mass media to understand, also made the ETS more vulnerable to regulatory capture [11]. Comparing the EU Green Paper [1] and the directive three years later [2], Markussen and Svendsen [119] conclude that industry lobbying succeeded in shifting several questions to their advantage, notably the question of deciding the amounts of free allocation at the Member State level. The resulting competition between countries led to over-allocated free allowances, high reserves for new entrants and higher flexibility by borrowing and linking with Kyoto flexible mechanisms.

Consequently, technical complexity can be considered as a disadvantage of the EU ETS, certainly when compared to a carbon tax.

6.3. The ambiguous effect of free allocation on political acceptability

During the first 2 periods of the EU ETS, allocation was almost completely free, politically motivated by carbon leakage. However, most studies on carbon leakage find that very few sectors are exposed to leakage and that even for these sectors 100% free allocation is not indicated (see above).

Besides, grandfathered free allocation is not a flawless way to avoid leakage or offshoring. Insofar as carbon costs are passed through, market share is still lost and leakage is not avoided, despite high profit margins. This explains why all *ex ante* studies on carbon leakage find leakage rates despite 100% free allocation. Moreover, higher prices create the same windfall profits for importers. Profits and competitiveness are not synonymous. They can be opposites, if higher product prices generate profits from free allocation but attract imports [[120], p. 22]. Output-based free allowances avoid the problem of cost pass-through. If carbon costs are not passed through, leakage is avoided, but with another efficiency loss: no internalisation of climate damage in the price and thus no demand-driven abatement.

Free allocation is politically justified for avoiding carbon leakage. But its massive use despite its imperfect effectiveness against leakage, windfall profits and distortional effects indicates that it has been strongly influenced by industrial lobbying. The significant revenue for many ETS producers made ETS more acceptable to industry than was a carbon tax. In fact, free allocation has the political advantage that it can help buy the support of some key industries that would otherwise oppose the plan [3]. Free allocation nevertheless entails a trade-off between political acceptability for industry, on one hand, and for other stakeholders on the other hand. Indeed the windfall profits induced by free allocation constituted a transfer of wealth from consumers to industry shareholders. Free allocation also impeded measures to compensate for the regressive effect of a price on carbon. This explains why the EU ETS failed to gain support from the wider public and environmental NGOs [121,122]. The gradual shift to auctioning during the 3rd period will therefore be a requisite to improve the political acceptability of the ETS for the population at large.

The way to full auctioning however, may be hindered again by regulatory capture. The new directive [7] states that, for the 2013–2020 period, sectors with more than 5% cost increase (direct and indirect cost of €30/t as a proportion of gross value added) and with more than 10% non-EU trade intensity (total value of non-EU exports + imports as a proportion of total EU market size) are considered to be at significant risk of carbon leakage and will receive up to 100% of grandfathered allocation. According to Clo [123] this is the case of only 6 out of 257 ETS sectors (NACE 4-digit sectors). Moreover, sectors with a trade intensity exceeding 30% as a single criterion (131 sectors) or a cost increase exceeding 30% as a single criterion (3 sectors) will receive 100% free allowances too. Clo [123] points out that there is no economic argument to consider these two criteria separately and concludes that 100% free allocation is not justified for the majority of these sectors. A sector with a high cost increase but negligible trade exposure can pass the extra cost through to consumers and a sector with high trade exposure but a negligible cost increase will not risk carbon leakage. Dröge and Cooper [124] state that additional criteria should have been selected for more accurately determining which sectors were at risk: cost structures, pass-through ability, abatement potential and institutional factors.

7. Conclusion

7.1. Environmental effectiveness

The EU ETS succeeded in reducing the emissions of the sectors covered. During the first phase, abatement compared to business as usual is estimated between 2.5% and 5%. During the second phase, the emissions cap will be 6.5% below 2005 levels. Thereby, the ETS will very likely succeed in imposing a proportional or more-than-proportional abatement from polluting sectors and effectively reach the Kyoto target. This is a major aspect of the environmental effectiveness of the policy. The third phase foresees a cap of 21% below 2005 emissions. Note that an important part of this abatement (the equivalent of 100% from the 2nd period and 33% from the 3rd period) is likely to be realised in developing countries and Eastern countries via Kyoto credits. The equivalence of European abatement and offsetting credits is controversial, but goes beyond the scope of this review.

The ETS did not lead to a meaningful increase in carbon emissions outside the EU: carbon leakage was lower than observable during the first phase. However, leakage should continue to be monitored and is an important future research question, as it is difficult to observe over a short period of time and because free allocation, which is the present anti-leakage measure, has many detrimental side-effects and will diminish in the future.

Over-allocation during the first phase, resulting in a carbon price crash in April 2006, was estimated around 6% of total allowances. Over-allocation also decreased the transparency of the environmental effectiveness, as it was justified in NAPs by upward-biased historical emissions and upward-biased Business-As-Usual estimations.

The environmental predictability of the policy was satisfactory as the future emissions were established by the cap. This environmental predictability is at the expense of a low predictability of abatement effort. Due to the present economic crisis, even if emissions are on track, the carbon price crashed below 7 euro, seriously dampening the economic effort to reduce emissions. The recent price crash indicates the need for future research on a carbon price floor, which could combine the advantages of carbon trading and a carbon tax.

Altogether, the EU ETS has scored well on environmental effectiveness because it succeeded in creating a historic trend break to lower aggregate emissions and to reach the environmental Kyoto targets for carbon-intensive industry.

7.2. Economic efficiency

The EU ETS was not cost-efficient in the sense that the cap was not stringent enough to induce a marginal cost of abatement equal to the marginal social cost of carbon. However, the cost-efficiency criterion is difficult to assess because the marginal social cost of carbon is very difficult to estimate and different renowned authors come up with values between \$7.4 and \$85/tonne of CO_2 equivalent.

Cost-effectiveness, easier to measure than cost-efficiency, was high because the carbon market had instated a common price among the 27 EU Member States, equalising marginal abatement costs and thus attaining a given abatement goal at least cost. However, the fact that updating of free allocation was officially, unofficially or presumed to be correlated to previous emissions created an incentive to postpone abatement. In most NAPs, emissions were proportional to output, which avoids the incentive to postpone abatement but fails to internalise carbon costs in sales price and thus prevents demand-driven abatement. Free allocation for new entrants creates the incentive of an investment subsidy and withdrawal of free allocation for closures creates the incentive to postpone closure of old installations, resulting in a higher than Pareto-efficient production capacity for polluting industries.

Transaction costs were reasonably low for big companies, but noticeably hampered cost-effectiveness for smaller companies. Transaction costs of a cap-and-trade system are higher than for its closest alternative, a carbon tax.

The dynamic efficiency of the EU ETS is mainly driven by its propensity to boost technology development. The high volatility of the carbon price – compared to the price certainty of a carbon tax – increases the risk profile of low-carbon investments, thereby increasing the cost of capital for these investments. As such, the price volatility of the EU ETS hampers technology development. Several authors mention that a price floor would be desirable to reduce risk for long-term green investments. However, the plausible assumption of increasing carbon prices over time, which is less likely under a carbon tax, favours long-term strategic positioning on low-carbon technologies.

The inherent propensity of the carbon market to focus on low-cost abatement opportunities induces a lower-than-dynamic-efficient incentive for high-end technologies that are expensive today but that have a high potential to become major solutions in the future. Other policy measures are needed to boost these technologies.

To conclude on the overall economic efficiency of the EU ETS, cost-efficiency could have been much higher by making the cap more stringent, but given the lower-than-efficient abatement goal, the ETS scored rather well on the short-term cost-efficiency criterion, notwithstanding secondary shortfalls induced by free allocation. It is insufficient as a single policy to push through low-carbon technologies. The interplay of the carbon market with other policies to stimulate technology development such as research subsidies, public pilot projects and a carbon price floor constitutes a fruitful future research area.

7.3. Distributional effects

The EU ETS scores poorly on the criterion of distributional effects. Windfall profits were high, particularly for electricity producers, transferring billions of euros from (intermediary and final) consumers to shareholders. Internalising the social cost of carbon in sales prices is regressive because consumption within low-income households is more carbon-intensive. This regressive effect was not compensated for. Intercountry transfers were reasonably low and intersectorial transfers were considerable, but not very relevant as the only net buying sector – electricity – made the greatest windfall profits. It can be concluded that distributional effects were a major drawback of the ETS in its present form. A comprehensive study on transfers between low-income and high-income households induced by the EU ETS would be welcome to make the discussion about compensations more concrete.

7.4. Institutional feasibility

Institutional feasibility of the EU ETS was high: it gained support from major industrial groups as well as several major NGOs, and neutralised resistance of climate-unambitious business-lobbying groups and Member States, who blocked an earlier attempt to instate a European-wide carbon tax. However, business support for the EU ETS was founded on almost 100% free allocation, making the EU ETS extremely profitable for the majority of participants. The losers in these distributional effects, mainly electricity consumers and low-income households, had a lower weight in the political balance. This was, among other things, because the distributional effects of a cap-and-trade system are difficult for the average person to understand and

difficult for academics to quantify. In the end, shifting from free allocation to auctioning, as is planned from 2013, will be beneficial for long-term democratic support of the EU ETS.

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